Thesis/
Reports
Pilliod,
D.S.

POST-FIRE RECOVERY OF STREAM AMPHIBIANS, BENTHIC MACROINVERTEBRATES, AND RIPARIAN VEGETATION IN A FEDERALLY DESIGNATED WILDERNESS

06-JV-11221644-175

Post-fire Recovery of Stream Amphibians, Benthic Macroinvertebrates, and Riparian Vegetation in a Federally Designated Wilderness

Final Report

Submitted 19 February 2008

2/19/2008

Federal Contract between
Cal Poly Corporation of the California Polytechnic State University
And
USDA Forest Service, Rocky Mountain Research Station

RMRS Library
USDA Forest Service
MAR 14 2008
240 W Prospect Rd
Fort Collins CO 80526

GDO Award Number: 06-236
Cal Poly Project Number: 48200

Principal Investigators:
David S. Pilliod[1,2]
Robert S. Arkle[1,2]
Michael A. Yoshimura[1]

[1]Department of Biological Sciences, California Polytechnic State University, San Luis Obispo, CA, USA
[2] Present address: U.S. Geological Survey, Forest and Rangeland Ecosystem Science Center, 970 Lusk St., Boise, ID 83706, USA. E-mail: dpilliod@usgs.gov and rarkle@usgs.gov

SUMMARY

This is the final report for the Federal Contract 06-236 "Post-fire recovery of stream amphibians, benthic macroinvertebrates, and riparian vegetation in a federally designated wilderness" between the USDA Forest Service and Cal Poly Corporation. To examine linkages between upland, riparian, and instream post-fire recovery patterns, we compared habitat and biotic conditions in several burned watersheds two and six years post-fire. Six years after the Diamond Peak Fire Complex of 2000, we found that many habitat characteristics within the burned watersheds did not exhibit a trend towards recovery. Remotely sensed reflectance values showed some watershed-level vegetation recovery one year post-fire, but little subsequent change from two to six years post-fire. From two to six years post-fire, upland litter, moss, grass, and shrubs increased significantly, whereas tree densities decreased substantially within vegetation plots. No tree seedling or sapling recruits were observed in vegetation plots two or six years post-fire. This indicates that either the seed bank was consumed during the wildland fire, or that post-fire conditions are not suitable for seed germination. Riparian canopy cover over streams increased and water temperatures decreased significantly between two and six years after the fire. Instream large wood debris did not change between years, but in three of five streams, small organic debris decreased from 2002 levels. Most streams exhibited an increase in average substrate particle size and a decrease in average substrate embeddedness. Current velocity, discharge, and the annual peak streamflow were significantly higher in 2006 than in 2002 as a result of different

weather patterns. These high peak streamflows were likely important in influencing habitat and biotic parameters in the post-fire environment. Fish occupancy declined from 2002 to 2006 by ≥ 50 percent in four of five streams. A similar decline was observed in 2003 after high peak streamflows. Tailed frog larval densities decreased significantly in three of five streams. Much of this decline is due to a lack of one year old individuals, indicating that either reproduction by adults or survivorship of first year larvae was reduced in the burned watersheds. Macroinvertebrate community composition in the burned watersheds did not change significantly, but differed significantly from the unburned reference watersheds both two and six years post-fire. Our results indicate that six years post-fire, little recovery has taken place in the upland vegetation community. Riparian vegetation has regenerated more quickly and may play a role in stabilizing several instream conditions. While some instream habitat characteristics appear to be recovering, other habitat parameters, fish, amphibians, and benthic macroinvertebrates do not appear to have stabilized. These findings support the notion that although many native species can reestablish or maintain populations following disturbance, environmental variability can continue to affect community dynamics for several years post-fire, making recovery a slow and variable process.

INTRODUCTION

Few studies have examined how post-fire recovery in upland and riparian communities influences recovery in stream communities. Because of the strong linkages between upland, riparian, and stream communities within a watershed, dynamics in stream communities in the post-fire environment are likely influenced by the magnitude of upland disturbance and the rate of revegetation in riparian and upland areas (Bisson et al. 2003). Over time, regeneration (e.g., seed and root sprouting) and increased ground cover (e.g., needle cast, grasses and forbs) in riparian and upland habitats are likely to slow water runoff, reduce erosion rates (sediment delivery), and decrease light availability in or adjacent to streams within burned watersheds. These environmental changes can result in changes in stream habitat conditions and communities over time.

Post-fire Changes in Upland Forests

In ponderosa pine dominated forests, post-fire tree regeneration is dependent on the burn severity of both the canopy and the ground, as severities in these two strata effect seed production and seed survivorship, respectively. Regeneration of the understory is typically much faster and can influence tree seedling establishment in burned forests, as well as affect the survivorship of seedlings present during the fire (Bonnet et al. 2005). Seedling regeneration may vary with burn severity. For example, three years after a fire in a ponderosa pine forest, seedling densities of 612, 450, and 0 seedlings·ha^{-1} were observed in patches of low, moderate, and high burn severity, respectively (Lentile et al. 2005).

Understory vegetation, including mosses, grasses, forbs, and shrubs, plays an important role regeneration of soil organics consumed by the fire, which influences pattern of erosion. Neary et al. (2005) and Stednick (1996) found that removing 20–40% of vegetative cover can result in alterations of hydraulic processes within a watershed. Dead plant matter, such as fine litter and coarse woody debris (CWD), aids in preventing down slope transport of eroded

material (Robichaud 2000, 2005). Litter and small CWD (< 7 cm diameter) on the ground are typically consumed by fires of any burn severity, whereas "future" small CWD in the form of tree branches, is typically consumed by crown or high severity fires. Thus post-fire small CWD levels often depend on fire characteristics and burn severities, but generally increase over time once trees begin to regenerate (Ferguson and Elkie 2003). Large CWD (> 7 cm diameter) generally increases with time after a fire, as snags decay and fall to the ground (Passovory and Fulé 2006).

Post-fire Changes in Riparian Areas

Riparian forests generally regenerate more quickly than adjacent upland communities because of their close proximity to a permanent water supply and because the nitrogen-fixing alder trees (*Alnus* sp.) that dominate this community can grow in and enrich nutrient poor soils often left after severe fires (Neary et al. 1999). Riparian forests are susceptible to flood and debris flow disturbances following wildland fire, which influences post-fire successional processes (Dwire and Kauffman 2003). Survival and regeneration adaptations to fluvial disturbance contribute to rapid post-fire recovery in riparian forests (Dwire and Kauffman 2003). The high degree of shared past fire disturbances between upland forests and adjacent riparian forests in the Pacific Northwest (Everett et al. 2003, Olsen and Agee 2005) raises the possibility that these adaptations are also a response to the historic selective pressures of wildland fire (Pettit and Naiman 2007).

Post-fire riparian forest regeneration rates depend on burn severity (Arkle and Pilliod, unpublished data), extent, and spatial configuration (Pettit and Naiman 2007). However, regeneration of riparian communities can be influenced by recovery in upland habitats, as disturbances (runoff, landslides, tree falls) in upland communities often affect riparian forests (Dwire and Kauffman 2003). Burning and subsequent reestablishment of riparian vegetation has short- and long-term influences on many physical, chemical, and biological components of instream ecosystems.

Post-fire Changes in Streams

Recovery of headwater streams in burned ponderosa pine forests is highly variable. This variability is a function of two factors: (1) the severity and extent of the fire, in particular, soil heating and canopy loss in riparian and upland areas; and (2) the unpredictable nature of precipitation events following the fire. In other words, the severity of fires and timing and intensity of post-fire rainfall interact to drive post-fire hydrological patterns and effects on stream habitats and communities (Gresswell 1999, Robichaud et al. 2000). Erosion rates, for example, in high severity burns, can increase by one or more orders of magnitude (Robichaud 2000, 2005). High severity burns may also be more likely to cause widespread loss of the riparian canopy, resulting in reduced shading in riparian areas and increased stream water temperatures (Amaranthus et al. 1989). Loss of riparian vegetation can also result in decreased sediment buffering capacity, changes in large woody debris dynamics, and alterations to stream bank structure.

It has been suggested that fire effects on stream macroinvertebrate communities initiate when 25-50% of the watershed is burned, depending on climate and topography (Minshall 2003). Macroinvertebrate communities have been shown to respond variably to fire, depending on pre-fire stream conditions and time (Mihuc and Minshall 1995, Minshall et al. 2001a,b,c, Minshall 2003). Immediately after a fire, densities of some species are reduced, but then recover within a year or two, whereas the stability of community structure and diversity are affected over a longer period of time (Minshall 2003, Vieira et al. 2004). For example, community composition and taxa richness have been found to return to pre-fire levels relatively quickly, but may be highly variable from year-to-year for five to ten years post-fire (Minshall et al. 2001b,c). This variability in community composition is often the result of increases and decreases in disturbance-adapted taxa, which typically have strong dispersal capabilities and may be tolerant of post-fire stream conditions such as increased sediment, higher water temperatures, and variable streamflows. Many stream-dwelling amphibians and salmonids are sensitive to increased water temperature, sedimentation, and extreme runoff events following wildland fire disturbance or removal of riparian vegetation (Rieman and Clayton1997, Pilliod et al. 2003, Dunham et al 2003, Bury 2008), but many of these species are adapted to changing habitat conditions and populations can recolonize or recover quickly following wildland fire (Rieman and Clayton 1997).

Objectives

The objective of this study was to determine how post-fire changes in upland forests and riparian forests influence stream habitats and communities from two to six years after a mixed-severity wildland fire in a ponderosa pine forest in a central Idaho wilderness area. We examined post-fire changes the environment and biotic communities in the context of post-fire succession and recovery.

METHODS

Study area

Our study streams were low-order (1–3), high-gradient streams located in the Big Creek drainage of the Middle Fork of the Salmon River (SF Salmon) in central Idaho (Figure 1; 44°57'N, 115°41'W). Elevations of sampled watersheds ranged from 1211 to 2711 m in the Big Creek drainage. Geological parent materials are predominantly decomposed granite derived from the Idaho Batholith. Stream flows are driven by snowmelt in late spring and early summer (May and June), with base flows occurring from July through September. Plant communities in all watersheds were similar, with upland vegetation dominated by subalpine fir (*Abies lasiocarpa*), Douglas-fir (*Pseudotsuga menziesii*), and Engelmann spruce (*Picea engelmannii*) at higher elevations and on north-facing slopes, and by ponderosa pine (*Pinus ponderosa*) and sagebrush (*Artemisia* sp.)-grass communities on south-facing slopes. Riparian forests were characterized by gray alder (*Alnus incana*), red osier dogwood (*Cornus stolonifera*), Rocky Mountain maple (*Acer glabrum*), willow (*Salix* sp.) and water birch (*Betula occidentalis*).

Big Creek is largely confined to the Frank Church-River of No Return Wilderness Area, and wildland fires are not currently suppressed under a "wildland fire use" policy. Consequently,

many watersheds within the Big Creek drainage burned in 1988, 2000, 2006, and 2007 wildfires. In August and September 2000, the Diamond Peak wildfire complex burned 606.1 km^2 of the Big Creek and Middle Fork of the Salmon River drainages. This mixed severity wildland fire burned portions of both upland and riparian forests in many watersheds.

Study design

To examine the associations between watershed-level burn severity and recovery of stream habitat, amphibians, and macroinvertebrate communities, we sampled and compared tributaries of Big Creek in 2002 and 2006, following the Diamond Peak Fire of 2000. Within these watersheds, we compared disturbance effects on habitat and amphibians (n = 5 streams, 30 transects per stream per year) and macroinvertebrate community composition (n = 6 streams, five Surber samples per stream per year) across increasing riparian burn severity (Figure 1). Within the Cabin-Cow Creek watershed, we sampled 13 vegetation plots in 2002 and again in 2006 to document post-fire changes in upland habitats. We used remote sensing (Landsat multispectral satellite) to determine how reflectance values, measured as the normalized burn ratio (NBR), from entire watersheds (n = 8 burned and 8 unburned watersheds) changed from 1999 to 2006.

Remotely sensed habitat modeling

Burn severity was estimated at the watershed-level within the Big Creek drainage using normalized burn ratio values (NBR) calculated from Landsat 7 Enhanced Thematic Mapper satellite imagery following Key and Benson (2006). Delta NBR (ΔNBR) was calculated as the difference in NBR between available cloud-free, pre- and post-fire images taken on 18 July 1999 and 9 September 2001, respectively. We used published values (see Key and Benson 2006) to determine breakpoints in ΔNBR and assigned each pixel in a watershed as either unburned, low severity, or high severity (Figure 1). We then calculated the percentage of pixels in each watershed that fell into the high and low burn severity categories and summed these to estimate the total percentage of each watershed that burned. Finally, using a geographic information system (ArcGis 9.0, ESRI, Redlands, CA), we selected only those pixels within 20 m of a stream to calculate the percent of pixels in high (HIGHSEV) and low burn severity classes within the complete length of the riparian forest to estimate the relative riparian burn severity per watershed.

As upland and riparian communities regenerated after the 2000 fire, annual changes in reflectance values (NBR) were measured at the watershed level, using Landsat images of the study area. Within each watershed and year (1999 to 2006), NBR values for each 30 m pixel were averaged. Average NBR values from each burned watershed (where $\geq$ 20 percent of the watershed burned, n = 8) were compared with average NBR values from eight unburned reference watersheds from the SF Salmon drainage in each year.

Vegetation plot sampling

ArcGis 9.0 was used to generate a list of randomly selected GPS coordinates. In 2002, we navigated to each point and established a sampling plot. In each plot, we sampled three different strata: ground cover, understory vegetation, and tree. Within the ground cover stratum, we

sampled in a 2.07 m radius of the plot center (1/300th acre circular plot), making ocular estimates of percent cover of litter, moss, mineral soil, and brown cubic rotten wood (BCR). At each plot, we sampled small (< 7 cm diameter) and large (> 7 cm diameter) course woody debris (smCWD and lgCWD, respectively) in three 120 ft. transects measured out in the directions of: 0, 120, and 240 degrees. Within the understory vegetation stratum, we sampled the same 2.3 m radius (1/300th acre) circular plot for percent cover of grasses, forbs, low (< 45 cm), medium (45-183 cm), and tall (>183 cm) shrubs, and tree seedlings (< 2.54 cm diameter). The tree stratum was sampled in a 2.07 m radius (1/24th acre) plot on the same center as the other strata. We determined the density of saplings (2.54-12.44 cm diameter at breast height) and trees (>12.44 cm diameter at breast height). For trees, we also determined basal area, stand density index, crown competition factor, top height, mean diameter, and cubic feet per acre.

Stream and riparian habitat sampling

Each year (2002 and 2006) we sampled 30 randomly spaced, 1 m belt-transects within a 1 km reach of each stream. Transects that occurred in waterfall, pool, and log microhabitats were moved a random distance upstream. At each belt-transect, we recorded water temperature, wetted width, depth, current velocity, substrate size, substrate embeddedness. To characterize substrate size and embeddedness, we performed a Wolman Pebble Count (Wolman 1954) on 30 pebbles per belt-transect (900 pebbles per stream annually) using a modified substrate size class (Platts et al.1983). The embeddedness of each pebble was categorized as: 0-5, 6-25, 26-50, 51-75, or 76-100% embedded in fine sediment. We measured the percent of each belt-transect occupied by large woody debris (LWD; > 5 cm diameter) and organic debris (ORGANIC; wood < 5 cm diameter and leaf litter). We also recorded the percent of each belt transect covered by riparian vegetation (COVER) and by undercut banks (UNDERCUT).

Fish sampling

The presence or absence of fish (chinook salmon, *Oncorhynchus tsawytscha;* bull trout, *Salvelinus confluentus*; westslope cutthroat, *O. clarki lewisi*; steelhead/rainbow trout, *O. mykiss* spp; or sculpin, *Cottus* spp) was recorded for each belt-transect. We determined the proportion of 30 belt-transects occupied per stream and year and compared these values to determine changes in fish populations.

Amphibian sampling

In each belt-transect, we kick sampled for Rocky Mountain tailed frog larvae and adults. All large rocks and LWD were inspected for amphibians and removed from the transect area. Subsequently, the remaining substrate within the transect was dislodged to a depth of about 10 cm. Displaced tadpoles and adults were collected in D-frame nets at the downstream edge of the plot. For each stream, we calculated an average density from 30 transects sampled each year (2002 and 2006). These organisms provide good indicators of stream habitat conditions as they have long larval life stages (at least three years), and are known to be sensitive to sedimentation, high water temperatures (> 16 C), and LWD induced pool formation (Corn and Bury 1989,Welsh and Ollivier 1998, Pilliod et al 2003).

Benthic macroinvertebrate sampling

We collected benthic invertebrate samples from one location (the thalweg) along 5 transects per stream per year, using a standard Surber sampler (0.10 m^2, 500 μm mesh) during summer base flow conditions. In the laboratory, benthic macroinvertebrates were keyed to genus according to Merritt and Cummins 1996, except for Chironomidae and Oligochaeta, which were keyed to family. We calculated total density of all benthic macroinvertebrates (TOTALDEN), density of each taxa, percent Ephemeroptera, Plecoptera, and Trichoptera (EPT), and taxonomic richness (RICHNESS) for each sample.

Data analysis

To determine how habitat and biotic conditions in burned watersheds change through time, we tested for significant differences in response variables between watersheds and years of sampling (YEAR). Landsat, riparian, instream habitat, amphibian, and macroinvertebrate summary data were analyzed using general liner modeling (GLM) and a post-hoc Duncan's Multiple Range Test. Vegetation plot data were analyzed using matched pairs *t*-tests. Each of these univariate analyses were performed using SAS (version 9.0.1, SAS Institute 2006).

Watersheds that were sampled for benthic macroinvertebrates (n = 6 burned and 6 unburned) were grouped into three categories (BURNCLASS) on the basis of the burn severity of the riparian forest (HIGHSEV = percent of each watershed's riparian forest burned at high severity). Three watersheds where less than 7 percent of the riparian forest burned at high severity were grouped into the LOW category. Three watersheds where more than 7 percent of the riparian forest burned at high severity were grouped into the HIGH category. Six reference watersheds from the SF Salmon drainage were grouped into the Unburned category. These groups allow us to perform multivariate analyses of watersheds burned at different levels. We obtained the 7 percent high severity cutoff value by regressing the total percent of each watershed burned against the percent of each watershed's riparian forest burned at high severity. This cutoff value corresponds to a total burn percentage of 25, a value where effects on stream macroinvertebrates often initiate (Minshall 2003).

To test for multivariate (genera density) differences between benthic macroinvertebrate communities from each BURNCLASS and year, we first obtained average values for the density of each genus from the five Surber samples collected per stream per year. The average community composition from each stream and year were partitioned into six groups (3 levels of BURNCLASS by 2 levels of YEAR). Differences between all six groups were analyzed using Multi-Response Permutation Procedure (MRPP) with Sorenson distance in PC-ORD 5.11 (Mielke and Berry 2001, Biondini et al. 1985). Unlike MANOVA and related methods, this technique provides a nonparametric analysis that does not assume multivariate normality, linearity, and homogeneity of variances. MRPP provides a test statistic (*T*), with more negative values indicating greater multivariate differences between groups, an effect size statistic, *A* (0 to 1), with values of one indicating perfect homogeneity within groups and 0 indicating that within-group homogeneity is no stronger than expected by chance, and a *p*-value indicating whether the mean within-group distance is smaller than the distance expected from chance alone (McCune et al. 2002).

Macroinvertebrate community composition and relationships to habitat and treatment variables were analyzed using nonmetric multidimensional scaling (NMS) ordinations (Kruskal 1964, Mather 1976). NMS is perhaps the best ordination technique currently available for community data for several reasons. NMS does not assume linear species response to environmental gradients or to other species, as this response is rarely observed in community data sets. NMS avoids the "zero-truncation" problem, where the absence of a species from a sample unit provides no indication of how poor the environment is for that species and NMS has performed well with zero-rich data and communities with high beta diversity (McCune et al. 2002).

Using PC-ORD 5.11 software (McCune and Mefford 2006), an NMS ordination was performed on log (x + 1) transformed genera density data. In the analysis, the average community composition from each stream in each year was analyzed and plotted to show the community similarity between each stream and year, and how HIGHSEV and YEAR are related to macroinvertebrate community composition. Sorenson (Bray-Curtis) distance was used to measure dissimilarity between sample units (streams). Two hundred fifty runs of real data and 250 runs of randomized data were used to provide a Monte Carlo test of the statistical significance of each axis. The proportion of variance represented by each of the final dimensions was evaluated based on the correlation coefficient (r^2) between Sorenson distance in ordination space and original space. Linear relationships between community composition and reach/environmental variables were examined by correlations between these variables and ordination axes. In both analyses, genera present in < 5 percent of samples were excluded, as rare species can have undue influence on ordination results (McCune et al. 2002).

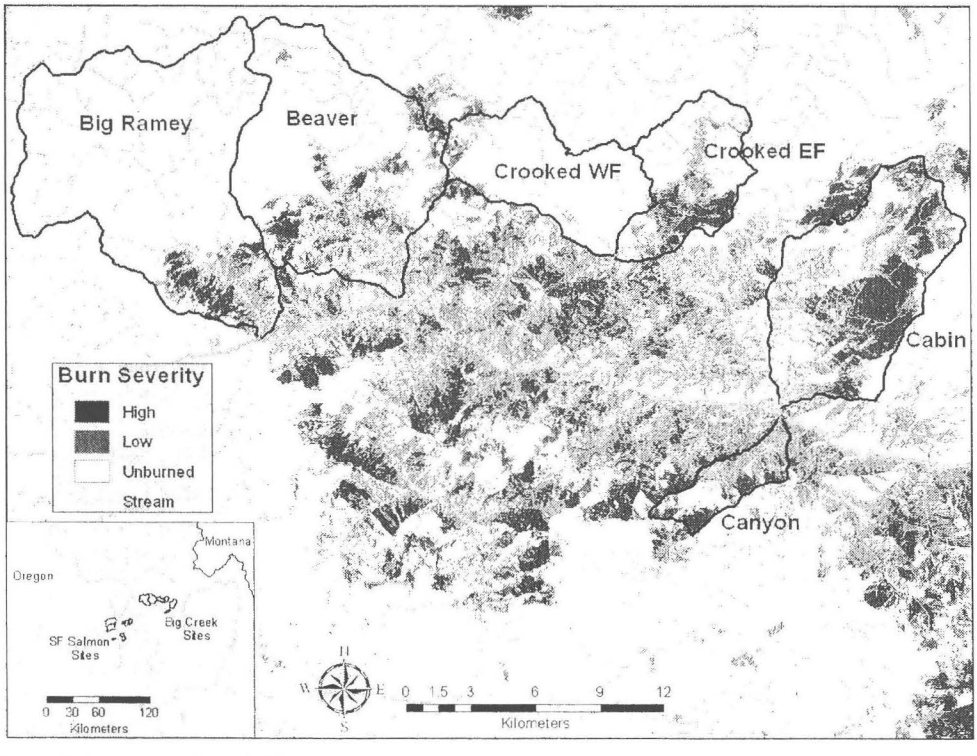

Figure 1. Burn severity within the Diamond Peak Fire Complex. Big Creek (burned) and SF Salmon (unburned) drainages showing delineations of each watershed sampled for benthic macroinvertebrates (inset).

RESULTS & CONCLUSIONS

Remotely sensed habitat modeling

In 2001, normalized burn ratio (NBR) values calculated from Landsat 7 imagery decreased significantly in Big Creek watersheds (n = 8) following the Diamond Peak Fire Complex (Figure 2). Post-fire (2001 – 2006) NBR values in unburned watersheds (n = 8) did not differ significantly from 1999 pre-fire values, however NBR values in 2000 were significantly higher than all others. Post-fire NBR values in the Big Creek watersheds were significantly lower than those from the SF Salmon watersheds through 2006. Within the Big Creek watersheds, there was no significant increase in NBR from 2001 to 2006.

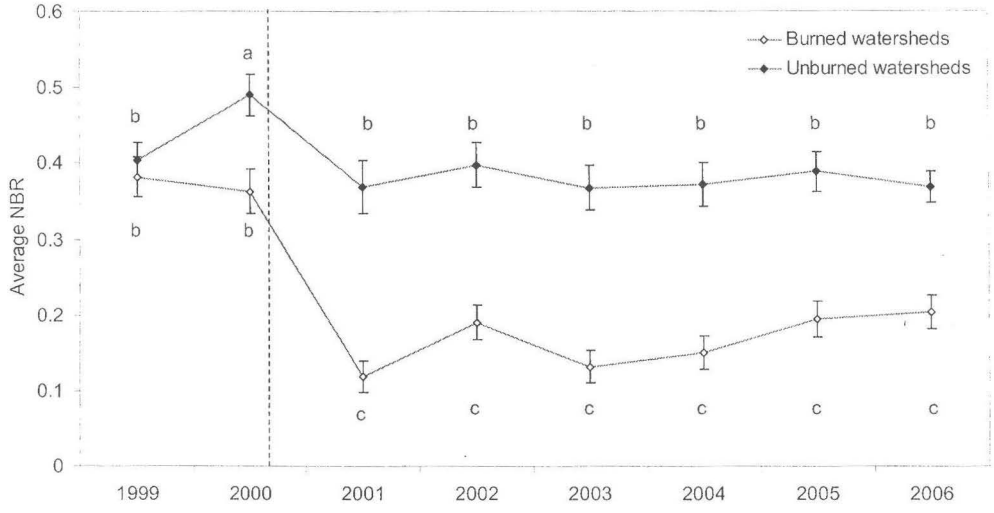

Figure 2. Average (±1 SE) normalized burn ratio (NBR) values for Big Creek watersheds (n = 8) burned in the 2000 Diamond Peak Fire Complex and for unburned SF Salmon watersheds (n = 8). Higher values of NBR correspond to increased green reflectance. Lower values of NBR correspond to more brown and black reflectance. Dashed line indicates timing of the wildland fire. Groups with the same letter are not significantly different (α = 0.05) according to Duncan's Multiple Range Test.

Vegetation plots

Within the ground stratum, litter cover, moss cover, large coarse woody debris (lgCWD; > 7 cm diameter), and total coarse woody debris (allCWD) increased significantly between 2002 and 2006 (Table 1). There was no change in mineral soil cover, brown cubic rotten wood (BCR), or small coarse woody debris (smCWD; < 7 cm diameter). Within the understory stratum, grass cover, low shrub cover (LowSHRUB; < 45 cm height), and medium shrub cover (MedSHRUB; 45-183 cm height) increased significantly, whereas forb and tall shrub cover (TallSHRUB; > 183 cm height) did not change from 2002 to 2006. Within the tree stratum, tree density decreased by an average of 65 trees per acre, indicating that many trees that were living two years post-fire had died and become snags by 2006, six years post-fire. No tree seedlings or saplings were found in any vegetation plot. Additionally, no seedlings or saplings were observed growing on any sloped terrain that burned in the 2000 wildland fire; however seedlings were observed on burned terrain that was relatively flat. This indicates that either the seed bank was consumed during the wildland fire, or that on xeric slopes, conditions are not suitable for seed germination.

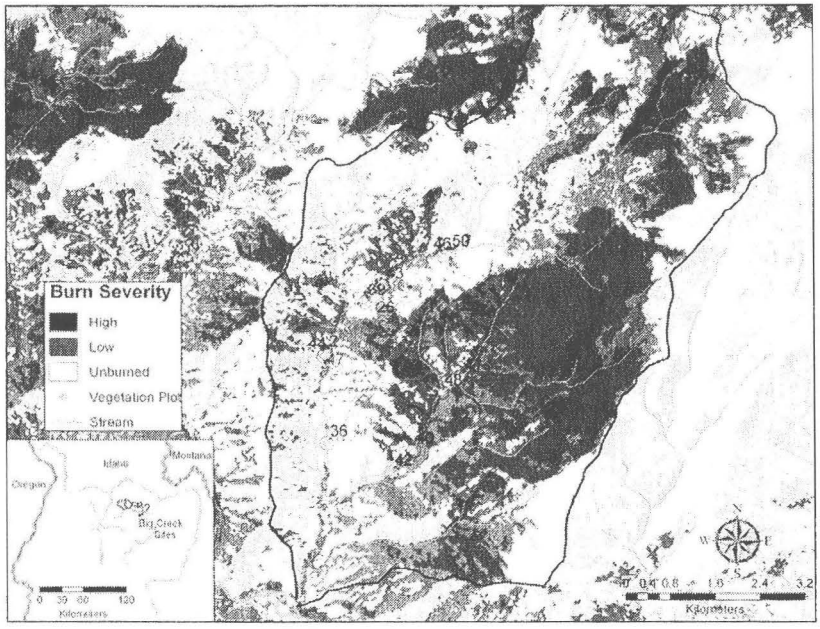

Figure 2. Location of vegetation plots sampled in 2002 and 2006 within the Cabin-Cow Creek watersheds. These plots were randomly chosen from all of the 2002 plots falling within 120 m of the riparian zone of Cabin or Cow Creek.

Table 1. Results of matched pairs t-test for variables from vegetation plots (n = 13) sampled in 2002 and resampled in 2006. Mean change represents the amount and directionality by which the variable changed in the average plot and SE represents the standard error of that mean. *p*-values are based on two-sided alternative hypotheses. To obtain *p*-values for directional hypotheses, divide by two.

Strata	Variable	Units	Mean change	SE	t-value	*p*-value
Ground	Mineral soil	% cover	+0.333	5.06	0.07	0.9487
	Litter	% cover	+54.2	12.1	4.48	0.0009***
	BCC	% cover	-0.667	0.752	-0.89	0.3944
	Moss	% cover	+10.1	1.71	5.79	0.0001***
	smCWD	tons/acre	+1.69	1.06	1.59	0.1386
	lgCWD	tons/acre	+3.94	1.51	2.62	0.0226*
	allCWD	tons/acre	+5.63	1.59	3.53	0.0041**
Understory	Grass	% cover	+14.8	5.52	2.63	0.0236*
	Forbs	% cover	-0.25	3.11	-0.08	0.9373
	LowSHRUB	% cover	+17.8	4.37	4.08	0.0018**
	MedSHRUB	% cover	+9.92	2.91	3.41	0.0059**
	TallSHRUB	% cover	-0.25	0.217	-1.15	0.2750
	Tree seedling density	ind/acre	0	0	na	na
Tree	Sappling density	ind/acre	0	0	na	na
	Tree density	ind/acre	-65.1	23.2	-2.84	0.0158*
	Basal area	ft^2/acre	-27.2	26.3	-1.04	0.3210
	Stand density index	none	-51.9	37.8	-1.37	0.1943
	Crown competition factor	none	-25.5	22.0	-1.16	0.2691
	Top height	ft	+8.23	13.5	0.61	0.5526
	Mean diameter	in	-4.18	3.09	-1.35	0.2019
	Wood volume	ft^3/acre	+1097	914	1.20	0.2534

* indicates $p < 0.05$, ** indicates $p < 0.01$, *** indicates $p < 0.001$

Stream and riparian habitat

Several stream habitat variables changed between 2002 and 2006. Water temperatures decreased significantly during times of sampling (1-10, August 2002 and 2006) in three of five streams (Figure 3), although streamside air temperatures also decreased slightly, making it difficult to determine if the decrease in water (and streamside air) temperature is due to recovery of the riparian canopy, or to differences in weather. Ambient air temperatures during 2006 sampling reached daily highs of 38 C, however streamside air temperatures averaged 19.8 C, indicating that riparian vegetation may be influencing streamside air and instream water temperatures.

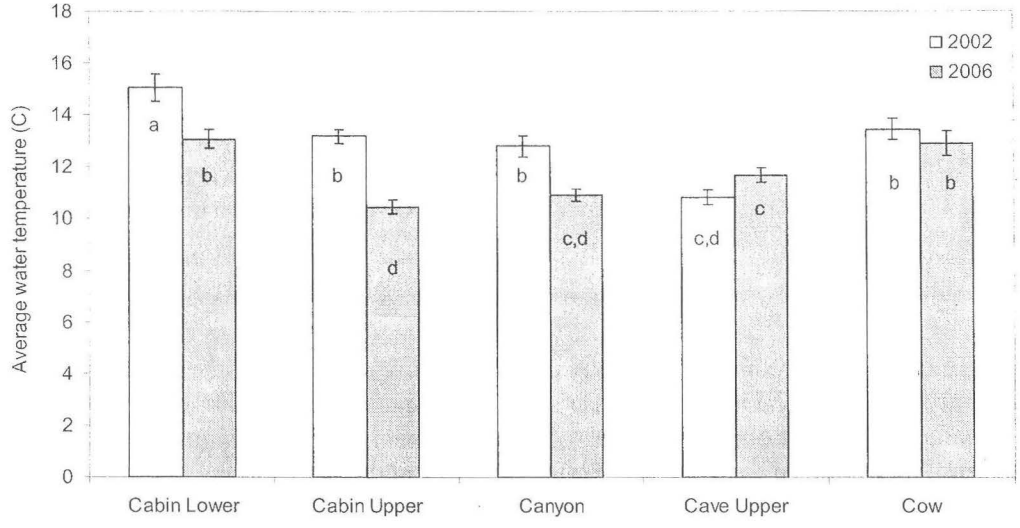

Figure 3. Average (±1 SE) water temperature recorded at the time of sampling each transect (n = 30) in 2002 and in 2006. Groups with the same letter are not significantly different ($\alpha = 0.05$) according to Duncan's Multiple Range Test.

Riparian canopy cover increased significantly in Canyon and Cow Creeks, the two watersheds with the most severely burned riparian forests (Figure 4). Canopy cover (COVER) in the other three watersheds increased slightly, although not significantly. This increase in canopy cover may contribute to the decline in water temperatures. In 2002, the average UNDERCUT in the two watersheds with the most severely burned riparian forests (Canyon and Cow Creeks), was significantly higher than in the other three watersheds. In 2006, the average UNDERCUT in Cow Creek was significantly higher, and the average UNDERCUT in Cave Upper was significantly lower, than in the other three watersheds.

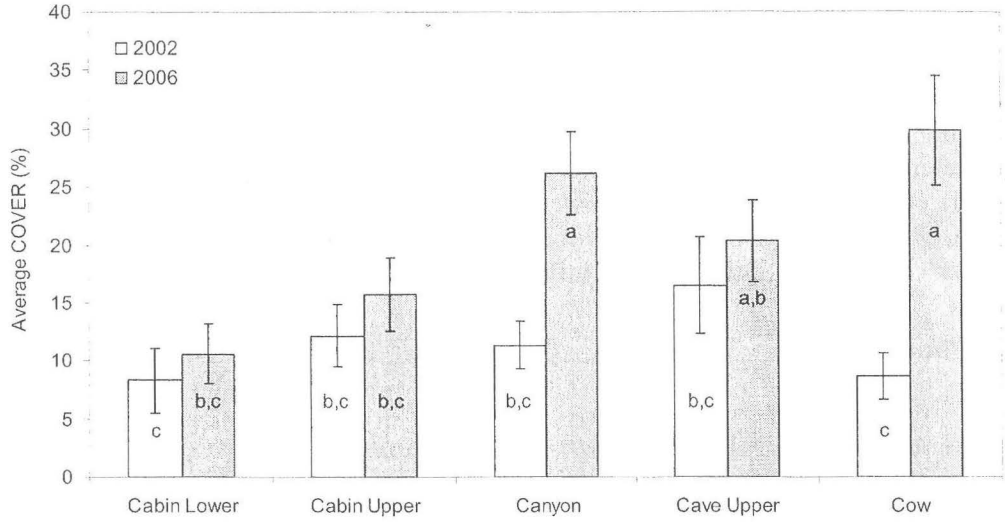

Figure 4. Average (±1 SE) percent of belt-transect (n = 30) covered by riparian tree canopy in 2002 and in 2006. Groups with the same letter are not significantly different ($\alpha = 0.05$) according to Duncan's Multiple Range Test.

The amount of instream large woody debris (> 5 cm diameter) did not change significantly from 2002 to 2006 in any of the five watersheds sampled. However, the volume of instream organic debris (all organic matter < 5 cm diameter) decreased significantly in Cabin Lower, Cabin Upper, and in Cow Creek (Figure 5). Organic debris volumes in Cave Upper and Canyon Creeks did not change significantly from 2002 to 2006.

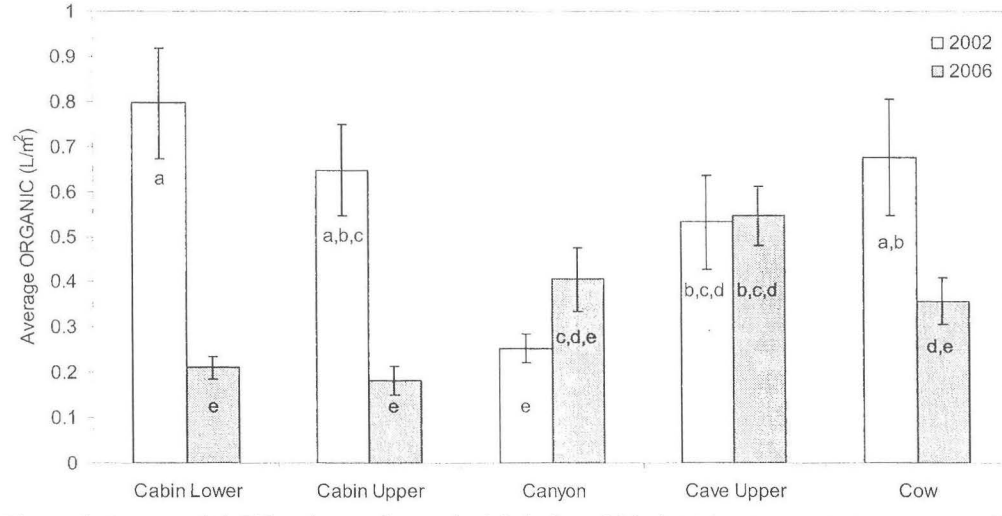

Figure 5. Average (±1 SE) volume of organic debris from 30 belt-transects per stream per year. Groups with the same letter are not significantly different ($\alpha = 0.05$) according to Duncan's Multiple Range Test.

The size of the average benthic substrate particle (Average pebble size) increased significantly in all streams except Cabin Lower and Cabin Upper (Figure 6). Larger streams had larger average pebble sizes, while the two smaller streams, Canyon and Cow, had relatively

smaller substrate particles. The average substrate embeddedness (percent of pebble embedded in sand or sediment) decreased in the more severely burned watersheds, indicating that less sediment is entering or being retained within these streams. Embeddedness did not change significantly in Cabin Upper or Cave Upper (Figure 7).

At the time of sampling, the average current velocity (m/s) and discharge (m^3/s) were significantly greater in 2006 than in 2002 for all five streams. The annual peak streamflow, which occurred in early June, at the nearby South Fork Salmon River gage was more than 2 times greater in 2006 than in 2002. The 2006 peak flow was the highest recorded since 1997. These differences in streamflow likely had an important influence on both the habitat and biotic communities in the burned watersheds.

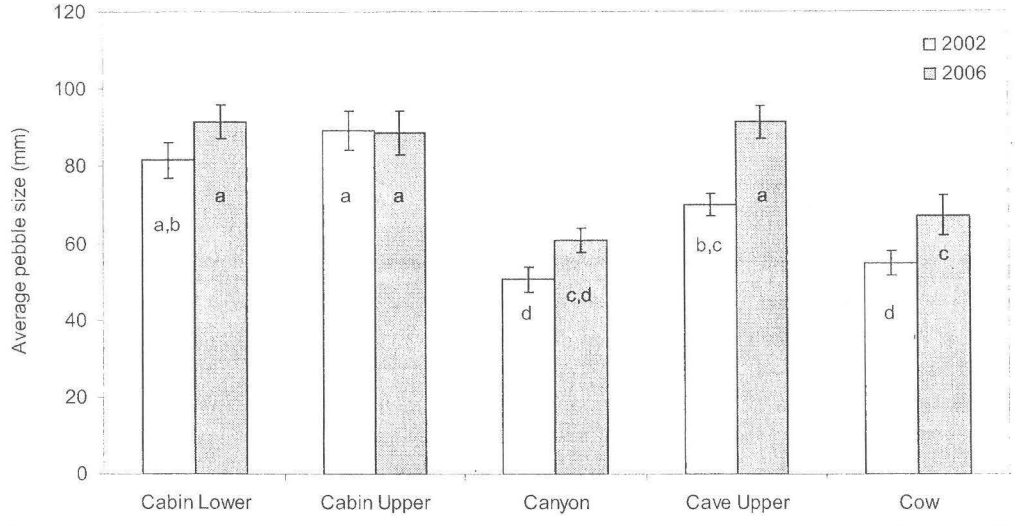

Figure 6. Average (±1 SE) pebble size from 30 pebbles measured (using Wolman pebble count) in 30 belt-transects per stream per year. Groups with the same letter are not significantly different ($\alpha = 0.05$) according to Duncan's Multiple Range Test.

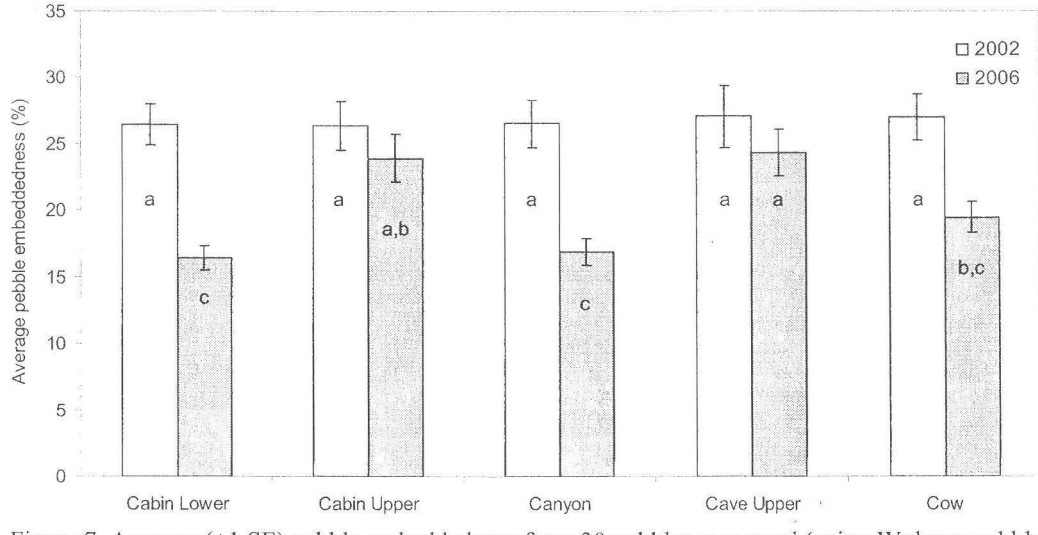

Figure 7. Average (±1 SE) pebble embeddedness from 30 pebbles measured (using Wolman pebble count) in 30 belt-transects per stream per year. Groups with the same letter are not significantly different ($\alpha = 0.05$) according to Duncan's Multiple Range Test.

Fish

The proportion of 30 belt-transects occupied by fish declined from 2002 to 2006 in all streams except for Cow Creek (Figure 8). A similar decline was observed in 2003 after high peak streamflows. This indicates that high peak streamflows in burned watersheds may reduce fish occupancy.

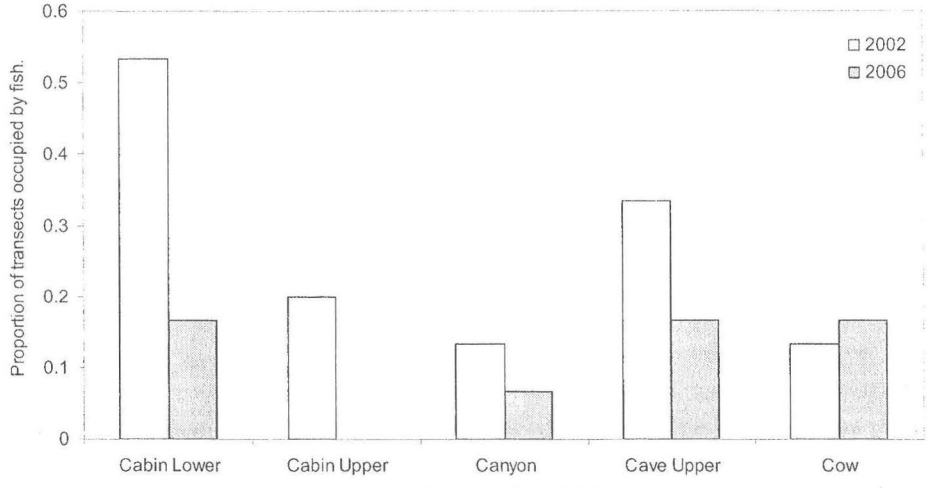

Figure8. Proportion of belt-transects occupied by fish (n = 30 belt-transects per stream per year).

Amphibians

Tailed frog larvae declined significantly in Cabin Lower, Cabin Upper, and Canyon Creeks (Figure 9). Much of this decline is caused by a lack of one year old individuals, indicating that either reproduction by adults or survivorship of first year larvae is reduced in the post-fire environment of these streams.

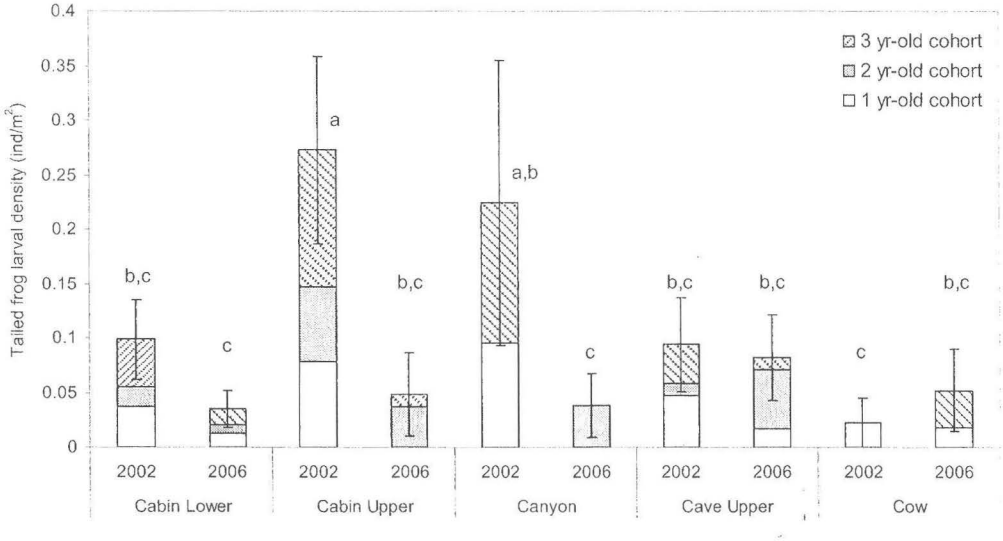

Figure 9. Average (±1 SE) total tailed frog larval density in 30 belt-transects per stream per year. Relative proportions of each cohort are shown. Total larval density in groups with the same letter is not significantly different ($\alpha = 0.05$) according to Duncan's Multiple Range Test.

Benthic macroinvertebrates

On the basis of the density of each genus, benthic macroinvertebrate communities in the HIGH and LOW groups were significantly different from the Unburned group in 2002 and continued to differ from the Unburned group in 2006 (Table 2). Macroinvertebrate communities in the HIGH and LOW groups were not significantly different from one another in 2002 or in 2006. The Unburned reference group had significantly different community compositions in 2002 and 2006, indicating that even unburned watersheds have annual changes in macroinvertebrate communities, probably due to variations in the magnitude of annual peak streamflow disturbance. These results indicate that rather than becoming more similar to unburned reference watersheds over time, the macroinvertebrate community composition in the burned watersheds is still significantly different six years post-fire.

Table 2. A- and *p*-values from MRPP pair-wise comparisons of macroinvertebrate community composition in each burn severity category. Densities of all 117 genera were averaged from 5 Surber samples per stream per year. The Unburned, LOW, and HIGH categories contain n = 6, 3, 3 watersheds respectively. *p*-values are not corrected for multiple comparisons.

		Unburned 2002		LOW 2002		HIGH 2002		Unburned 2006		LOW 2006	
		A	*p*	A	*p*	A	*p*	A	*p*	A	*p*
Unburned	2002										
LOW	2002	0.07	0.018*								
HIGH	2002	0.07	0.015*	0.06	0.163						
Unburned	2006	0.06	0.003**	0.14	0.002**	0.14	0.002**				
LOW	2006	0.06	0.039*	0.10	0.061	0.11	0.095	0.08	0.013*		
HIGH	2006	0.09	0.004**	0.09	0.034*	0.07	0.114	0.14	0.001**	0.02	0.331

* indicates $p < 0.05$, ** indicates $p < 0.01$

NMS ordination of macroinvertebrate genera densities from each stream and year combination shows the similarity of samples to one another and indicates which environmental variables are influencing the differences between communities (Figure 10). A three axis solution (final stress = 12.1, $p = 0.039, 0.019, 0.019$ for Axes 1, 2, 3, respectively) was chosen, representing 84.1% of the variance in the original data. This solution was reached after 41 iterations and resulted in a final instability of 0.00001. The biplot shows that in both years, streams cluster based on their burn severity category. These clusters fall along a gradient of HIGHSEV which is correlated with Axis 3 ($r^2 = 0.49$). The influence of year of sampling (2002 or 2006) is evident as streams also cluster by YEAR, which is correlated with Axis 1 ($r^2 = 0.40$). These results indicate that burn severity was influential in determining community composition in 2002 and continued to have an influence in 2006. Also, community composition within individual streams changed from 2002 to 2006 regardless of burn severity category.

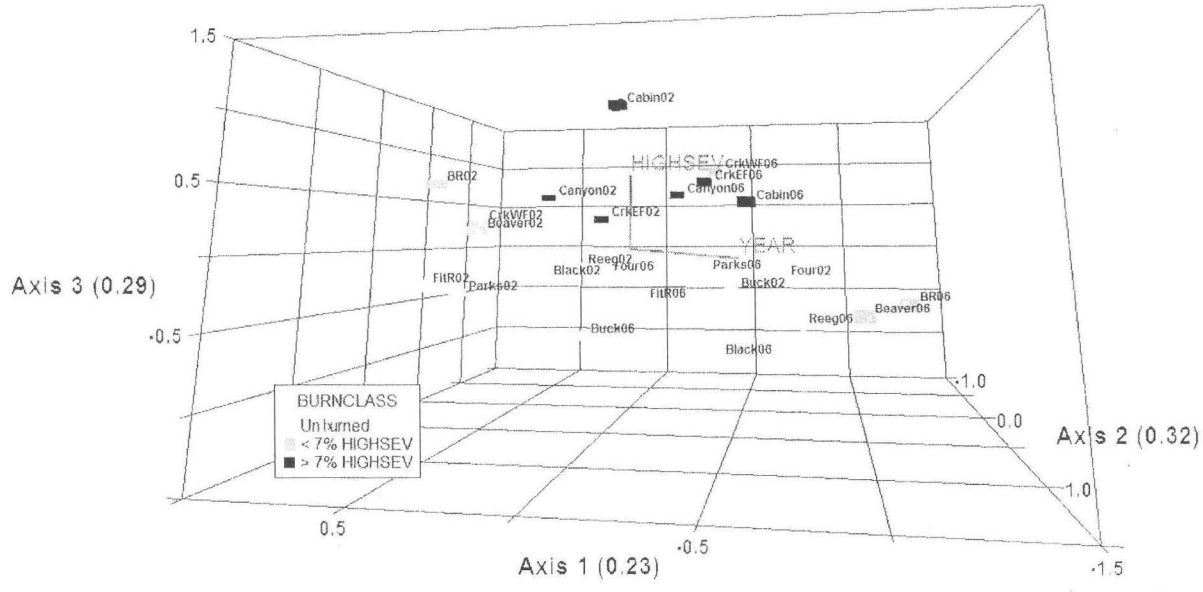

Figure 10. NMS ordination biplot of each stream's average (n = 5 Surber samples per stream per year) community composition plotted in three dimensional genera space. Streams plotted closer together are more similar based on community composition. The proportion of variance represented by each axis is included in parentheses. Sample point labels indicate the name of the stream and the year the sample was taken.

Macroinvertebrate total densities (TOTALDEN) declined significantly from 2002 to 2006 in all streams except Cabin Creek. Taxa richness (RICHNESS) also decreased significantly in all streams except Cabin Creek. These decreasing trends are likely the result of two factors; the reduction in periphyton production caused by increased canopy cover and decreased light availability, and the scouring effect of high peak streamflows in 2006.

ACKNOWLEDGMENTS

We thank the Krassel Ranger District, Payette National Forest for use of facilities and special use permits. All animal sampling was conducted under Idaho Department of Fish and Game Wildlife Collecting Permit #030716. Field data were collected by Robert Arkle, Jenny Liou, Annette Filice, Kelsey McDuffee, Dan Van der Elst, Jake Stoesz. Macroinvertebrate sorting and identification were provided by Annette Filice, Amanda Hiller, Casey Dilg.

LITERATURE CITED

Amaranthus, M., H. Jubas, and D. Arthur. 1989. Stream shading, summer streamflow and maximum water temperature following intense wildfire in headwater streams. In N.H. Berg (tech coord.), Proceedings of the symposium on fire and watershed management. USDA Forest Service General Technical Report PSW-109.

Biondini, M.E., C.D. Bonham, and E.F. Redente. 1985. Secondary successional patterns in a sagebrush (*Artemisia tridentata*) community as they relate to soil disturbance and soil biological activity. Plant Ecology 60(1): 25-36.

Bisson, P.A., B.E. Rieman, C. Luce, P.F. Hessburg, D.C. Lee, J.L. Kershner, G.H. Reeves, and R.E. Gresswell. 2003. Fire and aquatic ecosystems of the western USA: current knowledge and key questions. Forest Ecology and Management 178: 213-229.

Bonnet, V.H., A.W. Schoettle, and W.D. Shepperd. 2005. Postfire environmental conditions influence the spatial pattern of regeneration for *Pinus ponderosa*. Canadian Journal of Forestry Research 35:37-47.

Bury, R.B. 2008. Low thermal tolerances of stream amphibians in the Pacific Northwest: implications for riparian and forest management. Applied herpetology 5: 63-74.

Corn, P.S., and R.B. Bury. 1989. Logging in western Oregon: Response of headwater habitats and stream amphibian. Forest Ecology and Management 29(1-2): 39-57.

Dunham, J.B., M.K. Young, R.E. Gresswell, and B.E. Rieman. 2003. Effects of fire on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasions. Forest Ecology and Management 178: 183-196.

Dwire, K.A., and J.B. Kauffman. 2003. Fire and riparian ecosystems in landscapes of the western USA. Forest Ecology and Management 178: 61-74.

Everett, R., R. Schellhaas, P. Ohlson, D. Spurbeck, and D. Keenum. 2002. Continuity in fire disturbance between riparian and adjacent sideslope Douglas-fir forests. Forest Ecology and Management 175: 31-47.

Ferguson, S.H, and P.C. Elkie. 2003. Snag abundance 20, 30, and 40 years following fire and harvesting in boreal forests. The Forestry Chronicle 79(3): 541-549.

Gresswell, R.E. (1999). Fire and aquatic ecosystems in forested biomes of North America. Transactions of the American Fisheries Society 128: 193-221.

Key, C.H., and N.C. Benson. 2006. Landscape Assessment LA Sampling and Analysis Methods. USDA Forest Service General Technical Report RMRS-GTR-164-CD.

Kruskal, J.B. 1964. Nonmetric multidimensional scaling: a numerical method. Psychometrika 29: 115-129.

Lentile, L.B., F.W. Smith, and W.D. Shepperd. 2005. Patch structure, fire-scar formation, and tree regeneration in a large mixed-severity fire in the South Dakota Black Hills, USA. Canadian Journal of Forestry Research 35:2875-2885.

Mather, P.M. 1976. *Computational methods of multivariate analysis in physical geography.* J. Wiley and Sons, London.

McCune, B., J.B. Grace, and D.L. Urban. 2002. *Analysis of ecological communities.* MJM Software Design, Gleneden Beach, OR.

McCune, B, and M.J. Mefford. 2006. PC-ORD: Multivariate analysis of ecological data. Version 5.11. MjM Software, Gleneden Beach, OR.

Merritt, R.W., and K.W. Cummins. 1996. *An introduction to the aquatic insects of North America (Third edition).* Iowa: Kendall/Hunt Publishing.

Mielke, Jr, P.W., and K.J. Berry. 2001. *Permutation methods: A distance function approach.* NY: Springer series in statistics.

Mihuc, T.B., G.W. Minshall. 1995. Trophic generalists vs. trophic specialists: implications for food web dynamics in post-fire streams. Ecology 76: 2361-2372.

Minshall, G., J. Brock, D. Andrews, and C. Robinson. 2001a. Water quality, substratum and biotic responses of five central Idaho (USA) streams during the first year following the Mortar Creek fire. International Journal of Wildland Fire 10: 185-199.

Minshall, G.W., C.T. Robinson, D.E. Lawrence, D.A. Andrews, and J.T. Brock. 2001b. Benthic macroinvertebrate assemblages in five central Idaho (USA) streams over a 10-year period following disturbance by wildfire. International Journal of Wildland Fire 10: 201-213.

Minshall, G.W., T.V. Royer, and C.T. Robinson. 2001c. Response of the Cache Creek macroinvertebrates during the first 10 years following disturbance by the 1988 Yellowstone wildfires. Canadian Journal of Fish and Aquatic Sciences 58: 1077-1088.

Minshall, G.W. 2003. Responses of stream benthic macroinvertebrates to fire. Forest Ecology and Management 178: 155-161.

Neary, D.G., C.C. Klopatek, L.F. DeBano, and P.F. Folliott. 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management 122: 51-71.

Neary, D.G., K.C. Ryan, and L.F. DeBano. 2005. Wildland fire in ecosystems: effects of fire on soils and water. *Gen. Tech. Rep. RMRS-GTR-42-vol. 4.* Ogden, UT: USDA Forest Service, Rocky Mountain Research Station.

Olsen, D.L., and J.K. Agee. 2005. Historical fires in Douglas-fir dominated riparian forests of the southern Cascades, Oregon. Fire Ecology 1: 50-74.

Passovory, M.D, and P.Z. Fulé. 2005. Snag and woody debris dynamics following severe wildfires in northern Arizona ponderosa pine forests. Forest Ecology and Management 223: 237-246.

Pettit, N.E., and R.J. Naiman. 2007. Fire in the riparian zone: characteristics and ecological consequences. Ecosystems 10: 673-687.

Pilliod, D.S., R.B. Bury, E.J. Hyde, C.A. Pearl, and P.S. Corn. 2003. Fire and amphibians in North America. Forest Ecology and Management 178: 163-181.

Platts, W.S., W.F. Megahan, and W.G. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. General Technical Report INT-138, USDA Forest Service, Rocky Mountain Research Station. Ogden, UT.

Rieman, B., and J.C. Clayton. 1997. Wildfire and native fish: issues of forest health and conservation of sensitive species. Fisheries 22(11): 6-15.

Robichaud, P.R. Winter 2000. Forest fire effects on hillslope erosion: what we know. Watershed Management Council Networker, 9.

Robichaud, P.R., J.L. Beyers, and D.G. Neary. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. General. Technical. Report. RMRS-GTR-63. Fort Collins: USDA Forest Service, Rocky Mountain Research Station.

Robichaud, P.R. 2005. Measurement of post-fire hillslope erosion to evaluate and model rehabilitation treatment effectiveness and recovery. International Journal of Wildland Fire 14(4): 475-485.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology 176(1): 79-95.

Vieira, N.K.M., W.H. Clements, L.S. Guevara, and B.F. Jacobs. 2004. Resistance and resilience of stream insect communities to repeated hydrologic disturbances after a wildfire. Freshwater Biology 49: 1243-1259.

Welsh Jr., H.H., and L.M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: A case study from California's redwoods. Ecological Applications 8(4): 1118-1132.

Wolman, M.G. 1954. A method of sampling coarse river-bed material. American Geophysical Union, Transactions 35(6): 951–956.